Uncertainty Analysis in Ecological Risk Assessment: Sensitivity Analysis of Different Exposure Models

Ecological risk assessments are used for decision making regarding releasing new products to market as well as remedial action required to address contamination at legacy sites. Uncertainty analysis is considered a key part of the process (USEPA 1997) but assessments seldom address uncertainty quantitatively. Key sources of uncertainty include: Sampling and analytical variability (soil/sediment matrix, lab error), Choice of "Indicator" Species as Target Receptors for Different Exposure Pathways, Sample size. Bioaccumulation and Food Chain Modeling involve assumptions and uncertainty including: Linear assumption of bioaccumulation and selection of bioaccumulation factors; Other model inputs such as: Home range size, Dietary percentages, Body Weights, Ingestion Rate; Literature data on effects, Use of No Observed Adverse Effects Level (NOAEL) or Lowest Observed Adverse Effects Level (LOAEL) as a decision point. Toxicity testing where employed brings its own set of assumptions and interpretation that are subject to uncertainty. These include: Use of single organism, Correlation (or lack thereof) with Contaminants of Concern, Contaminant mixtures, and Reference area selection

Introduction

Another source of uncertainty affecting both food chain modeling and toxicity testing concerns whether the calculated risks and decisions made about them are based on individual or population impacts.

While all of these sources may be important in evaluating potential ecological risks, this paper focuses on the variability and resultant uncertainty associated with food chain modeling results used to predict risks from bioaccumulation. Food chain modeling is often used as the basis for derivation of site-specific remediation goals and hence is an important part of the ecological risk assessment and risk management process at legacy sites. The variability associated with food chain modeling inputs was assessed using test runs of different modeling scenarios to quantitatively evaluate results and conclusions drawn from them. Results and conclusions drawn from ecological risk assessments may vary widely depending on the choice of assumptions, degree of variability and whether or not site-specific empirical data are used as part of the assessment.

This paper focuses on one of the primary measures used to calculate ecological risks at hazardous waste sites in the United States, following USEPA (1997) guidance: food chain modeling. While ecological risk assessment studies may examine a variety of endpoints, this method is perhaps most used for development of remediation goals at contaminated sites for chemical waste products in soil and sediment.

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Food Chain Modeling Uncertainty

Food chain models typically used in the U.S. for assessment of ecological risks at legacy sites were used as a basis for examining the influence of different model inputs used to calculate risks.

Two target indicator species commonly used in aquatic and terrestrial ecological risk assessments in the U.S. were used as the basis for this study. The aquatic indicator species was the spotted sandpiper (*Actitis macularius*), a migratory shorebird species considered to be the most widespread sandpiper in North America (Cornell Lab 2024). This species is indicative of the sediment and surface water to benthos pathway, as the majority of its food items consist of aquatic and terrestrial insects (Bent 1929).

The terrestrial indicator species chosen was the American robin (*Turdus migratorius*), which is found over much of North America and is a common resident of yards and parklands (Cornell Lab 2024). It feeds primarily on soil dwelling invertebrates such as earthworms during summer months, while in the fall and winter it feeds mainly on fruit (Wheelwright 1986, 1988).

Food chain modeling compares the dose to toxicological effects or toxicity reference values (TRVs) and is expressed by the following formula (USEPA 1997):

$$HQ = \frac{\text{Dose}}{\text{TRV}}$$

Where:

HQ	=	Hazard Quotient
Dose	=	Exposure Point Concentration X Ingestion Rate / Body Weight
TRV	=	Toxicity Reference Value

The dose represents the intake rate of a contaminant, expressed as as mg/kg body weight per day. This is a function of the daily food ingestion rate for the target receptor being modeled, the amount of the COPEC in the soil present in the food item (represented by tissue results) and the incidental ingestion of soil.

The full equation for a single exposure pathway (e.g., soil to soil invertebrate to American robin) is given by **Equation 1** (adapted from NJDEP 2024):

$$HQ = \frac{\left(\frac{\left(EPC\ Soil\ \left(\frac{mg}{kg}\right) * BAF * IR\ \left(\frac{kg}{day}\right) * AUF\right) + \left(ISIR\ \left(\frac{kg}{day}\right) * EPC\ Soil\ \left(\frac{mg}{kg}\right)\right)}{BW\ (kg)}\right)}{TRV\ (\frac{mg}{kg}BW/day)}$$

Where:

EPC is exposure point concentration in soil (mg/kg);
BAF is bioaccumulation factor of the contaminant (proportion available for uptake, unitless);
IR is the ingestion rate of the animals (kg food ingested per day);
AUF is the area use factor (unitless; it is the home range in acres divided by the IAOC acres);

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The hazard quotient (HQ) is calculated by dividing the dose by a toxicity reference value (TRV), often taken from the literature, and often from a similar species but not necessarily from the exact indicator species being modeled. This is because wild species are much less often tested for toxicity than domestic species or those easily raised for use in laboratory tests. A hazard quotient greater than 1 is indicative of risk.

Typically risk estimates from food chain modeling are calculated using spreadsheets that summarize various assumptions as inputs. The inputs used for ingestion rate (IR), body weight (BW) and Area Use Factor (AUF) are generally taken from literature sources that in the U.S. are standardized (e.g. USEPA Exposure Factors Handbook, 1993). There is some uncertainty associated with the use of those values, in that obviously body weights vary by individual as do home range sizes and even ingestion rates. But in this case standardization of inputs is designed to obtain consistent population level estimates between investigations that apply to “average individuals” not effects as the organism level.

Exposure Point Concentrations (EPCs), Bioaccumulation Factors (BAFs) and Toxicity Reference Values (TRVs) may vary widely, however, and may be site-specific depending on chemical bioavailability. These inputs can greatly affect the assessment of ecological risks and the ultimate decision of whether remediation is required to address site risks.

Selection of Indicator Species

The indicator species chosen to reflect contaminant uptake may be an important determinant of risk calculations. For example, evaluating the soil invertebrate to avian pathway is a pretty standard means of assessing ecological risks. However, some locations (e.g. sandy soil environments) may support very low populations of soil invertebrates, such that modeling this pathway presents a very biased approach in evaluating ecological risks to that community. While this approach was not addressed in this paper, it is worth mentioning as a significant source of uncertainty in risk assessments.

In many cases the species used to evaluate a particular exposure pathway may not be reflective of actual site conditions. For example, while the Spotted sandpiper is often used as an indicator of aquatic risks, it may not be present at a given site, or if, so, only during part of the year. Similarly, habitat variables and quality can be important determinants of whether a species is present. In the case of the Spotted sandpiper shoreline habitats are preferred (Cornell 2024). If a site is thickly vegetated with shrubs for example, the shoreline may not be conducive to their use.

This paper does not go into the differences an investigator may obtain from using different indicator species, but it is worth noting before further discussing species-specific factors affecting the assessment.

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Home Range and Area Use Factor Uncertainty

While choosing home range size of an organism would be a straightforward process from the literature, territory and home range sizes may vary seasonally, with changes in population density, and in areas of different resource quality (Schoener, 1968, Sechaud et al 2022). As a result, home ranges reported in the literature may vary widely for the same species. Often "area use factors" are derived by dividing the site area by the home range to obtain an indication of the amount of time an organism spends at the site. Hence, what home range size is used is key in determining the degree of exposure and hence risks.

In practice the site area is often used as a basis of the area used by the organism since it provides "habitat", but in many cases habitat quality may vary greatly depending on the foraging and roosting characteristics of a species. Using the Spotted sandpiper as an example, one might take the area of a pond and divide it by the home range of the animal. One cannot say that a sandpiper would never use open areas of the pond, but in practice a sandpiper would spend most of its time along the shoreline and in shallow water areas so that mapping those would be a more accurate means of assessing risks to that indicator species. However, that is not usually the case in risk assessments that rely on conservative assumptions.

Methods

Sensitivity Analysis

To quantitatively evaluate the uncertainty associated with three major inputs (EPCs, BAFs, and TRVs), two typical ecological receptors were used. The first was the Spotted sandpiper (*Actitis macularius*), a widespread species representative of many wetland habitats in the U.S. and indicative of the sediment to benthic invertebrates to shorebird exposure pathway. The second was the American robin (*Turdus migratorius*), a common and widely distributed member of the thrush family that feeds largely on soil invertebrates throughout its breeding season (Wheelwright 1988).

To investigate the degree to which different assumptions may affect modeling results, different food chain modeling scenarios were evaluated, as summarized in **Table 1**.

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Table 1. *Food Chain Modeling Inputs Manipulated as Part of the Sensitivity Analysis*

Parameter	Contaminants in Sediment and Soil	Indicator Species	Variables
Exposure Point Concentration (mg/kg of contaminant in soil or sediment)	High molecular weight (HMW) Polycyclic Aromatic Hydrocarbons (PAHs) 4,4'-DDD 4,4'-DDE Copper Lead Mercury Methyl Mercury	Spotted sandpiper (aquatic); American robin (terrestrial)	Mean soil or sediment concentration; 95% Upper Confidence Level (UCL) in Soil or Sediment
Toxicity Reference Values (TRVs) (mg/kg BW/day per contaminant)	High molecular weight (HMW) Polycyclic Aromatic Hydrocarbons (PAHs) 4,4'-DDD 4,4'-DDE Copper Lead Mercury Methyl Mercury	Spotted sandpiper (aquatic); American robin (terrestrial)	Tier 1 TRV (NOAEL)[(1)] Tier 2 TRV (NOAEL) Tier 3 TRV (LOAEL)[(2)]
Bioaccumulation Factor (BAF)	Lead	Soil to Earthworms (for use in calculating risks to American robins)	Low BAF (site-specific tissue concentrations used directly); Medium BAF (from literature) High BAF (from literature)
Home Range/Area Use Factor	High molecular weight (HMW) Polycyclic Aromatic Hydrocarbons (PAHs) 4,4'-DDD 4,4'-DDE Copper Lead Mercury Methyl Mercury Zinc (sediment only)	Spotted sandpiper	Territory size (0.25 acre); 3 different Home Range estimates from the literature on breeding density

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Sensitivity Analysis of Exposure Point Concentrations

Exposure Point Concentrations (EPCs)

Exposure points concentrations (the amount of a contaminant to which a receptor is exposed) vary based on the number and distribution of samples collected. As a result, uncertainty may be very high in cases where the number of samples collected is low relative to the area under consideration. Uncertainty is also a function of the variability in the soil or sediment matrix (e.g grain size), as well as contaminant properties (e.g. propensity to adhere to organic carbon, lipid molecules, or volatilization near the surface). Risk calculations may be highly influenced by outliers, or "hotspot" locations since contaminant data is seldom normally distributed. In addition, the calculation of means or 95% upper confidence intervals that are often used as EPCs is affected by the presence of "non-detect" values, in which censored data (such as one half the detection limit) are often used. These values thus may have questionable statistical distributions since the variance is artificially constrained.

Typically risk assessments that rely on data with a high degree of variance have high uncertainty.

Exposure point concentrations were taken from a contaminated site in New Jersey, USA, as a generic example.

Table 2. *Contaminant concentrations in Surficial (0-6 ") Sediment at 1.08 Acre New Jersey Pond Site. (Shaded chemicals were evaluated using food chain modeling)*

Chemical	NJ Fresh Water Sediment Lowest Effects Level[1] (mg/kg)	Frequency of LEL Exceedance[7]			Maximum Value[9]	Mean[10] (w/ 1/2 NDs values)
Semi-Volatile Organic Compounds						
1,2-dichlorobenzene	0.294	0	/	33	0.084 U	N/A
1,3-dichlorobenzene	1.315	0	/	33	0.086 U	N/A
1,4-dichlorobenzene	0.318	0	/	33	0.097 U	N/A
1,2,4-trichlorobenzene	5.062	0	/	33	0.089 U	N/A
acenaphthene	0.00671	3	/	33	0.057	0.02
acenaphthylene	0.00587	7	/	33	0.24	0.04
anthracene	0.0572	15	/	33	0.81	0.11
benzo[a]anthracene	0.108	5	/	33	0.35	0.09
benzo[a]pyrene	0.15	2	/	33	0.22	0.08
benzo[b]fluoranthene	10.4	0	/	33	0.49	0.15
benzo[g,h,i]perylene	0.17	12	/	33	0.96	0.21

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benzo[k]fluoranthene	0.24	0	/	33	0.23	0.06
chrysene	0.166	15	/	33	1.6	0.26
dibenz[a,h]anthracene	0.033	10	/	33	0.41	0.06
fluoranthene	0.423	1	/	33	0.51	0.12
fluorene	0.0774	3	/	33	0.16	0.04
indeno[1,2,3-c,d]pyrene	0.2	3	/	33	0.35	0.11
naphthalene	0.176	5	/	33	0.82	0.15
phenanthrene	NC	0	/	33	0.75	0.14
pyrene	0.195	7	/	33	0.83	0.19
2-methylnaphthalene	0.0202	15	/	33	1.1	0.18
Low Molecular Weight PAH's with non-detects[3]	NC	0	/	33	2.69	0.64
High Molecular Weight PAH's with non-detects[4]	NC	0	/	33	3.555	0.81
Total Extractable Petroleum Hydrocarbons	NC	0	/	33	15000	2081
Pesticides						
aldrin	0.002	0	/	33	0.47 U	N/A
alpha-bhc	0.006	0	/	33	0.0058	0.04
alpha-chlordane	NC	0	/	33	0.019	0.04
beta-bhc	0.005	2	/	33	0.016	0.18
beta-chlordane	NC	0	/	33	0.69 U	N/A
4,4-DDD	0.00488	25	/	33	27	5.17
4,4-DDE	0.00316	21	/	33	2.9	0.62
4,4-DDT	0.00416	2	/	33	0.016	0.17
delta-bhc	NC	0	/	33	0.012	0.1
dieldrin	0.0019	0	/	33	0.91 U	N/A
endosulfan I	NC	0	/	33	0.61 U	N/A
endosulfan II	NC	0	/	33	3 U	N/A
endosulfan sulfate	0.0346	1	/	33	0.71	0.1
endrin	0.00222	0	/	33	1.9 U	N/A
endrin aldehyde	0.48	0	/	33	0.0054	0.08
endrin ketone	NC	0	/	33	1.7 U	N/A
gamma-BHC	0.003	1	/	33	0.0057	0.05
heptachlor	0.0006	2	/	33	0.0035	0.07
heptachlor epoxide	0.00247	2	/	33	0.041	0.04
methoxychlor	0.0136	0	/	33	5 U	N/A
toxaphene	0.000077	0	/	33	39 U	N/A
Metals and Inorganics						
aluminum	25500	4	/	33	30100	21288
antimony	NC	0	/	33	7.74	4.33

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arsenic	6	25	/	33	85.1	45.85
barium	NC	0	/	33	903	506
beryllium	NC	0	/	33	1.4	0.96
cadmium	0.6	25	/	33	12.9	4.74
calcium	NC	0	/	33	63600	23525
chromium	26	25	/	33	192	91.01
chromium VI	NC	0	/	33	43.5 U	N/A
cobalt	50	0	/	33	15.7	10.55
copper	16	25	/	33	376	213
iron	NC	0	/	33	64500	42988
lead	31	25	/	33	4200	1959
magnesium	NC	0	/	33	7870	5548
manganese	630	3	/	33	841	436
mercury	0.174	35	/	36	8.1	4.43
methyl mercury	NC	0	/	33	0.016	0.01
molybdenum	NC	0	/	33	8	6.5
nickel	16	25	/	33	58.2	41.35
potassium	NC	0	/	33	5110	3824
selenium	NC	0	/	33	7.53	3.81
silver	0.5	23	/	33	7.6	4.09
sodium	NC	0	/	33	3030	879
vanadium	NC	0	/	33	212	139
zinc	120	25	/	33	886	417
thallium	NC	0	/	33	4.67	2.4
cyanide (total)	0.1	8	/	33	21	1.74
extractable cyanide	NC	0	/	33	3 U	N/A

Notes: High Molecular Weight PAH's with non-detects calculated as a sum of fluoranthene, pyrene, benz(a)anthracene, chrysene, benzo(a)pyrene, dibenz(a,h) anthracene as provided in http://ceqg-rcqe.ccme.ca/download/en/243, Canadian Sediment Quality Guidelines for the Protection of Aquatic Life.

Half of MDL used for non-detected values. PAH's with non-detects calculated as a sum with non-detects as 1/2 of the MDL. N/A = Not Analyzed. NC = No Criteria Available. mg/kg = Milligrams per kilogram

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Table 3. *Contaminant Concentrations in Surficial Soil (0-6 “) at New Jersey Site*

Chemical	NJ ECO SOIL SSL (mg/kg)	Frequency of NJ ECO SOIL SSL Exceedance			Mean Value (w/ 1/2 NDs)	Maximum Value[9]
acetone	NC	0	/	139	0.16	1.4
benzene	0.255	0	/	139	0.01	0.083
bromochloromethane	NC	0	/	139	N/A	0.13 U
bromodichloromethane	0.54	0	/	139	N/A	0.13 U
bromoform	15.9	0	/	139	N/A	0.13 U
bromomethane	0.235	0	/	139	N/A	0.25 U
2-butanone	NC	0	/	139	0.02	0.075
carbon disulfide	NC	0	/	139	0	0.033
carbon tetrachloride	2.98	0	/	139	N/A	0.13 U
chlorobenzene	13.1	0	/	139	0	0.004
chloroethane	NC	0	/	139	N/A	0.25 U
chloroform	1.19	0	/	139	0	0.004
chloromethane	NC	0	/	139	N/A	0.25 U
cyclohexane	NC	0	/	139	0	0.028
1,2-dibromoethane	NC	0	/	139	N/A	0.13 U
1,2-dibromo-3-chloropropane	NC	0	/	139	N/A	0.25 U
dibromochloromethane	2.05	0	/	139	N/A	0.13 U
1,2-dichlorobenzene	2.96	0	/	139	0	0.001
dichlorodifluoromethane	NC	0	/	139	N/A	0.25 U
1,2-dichloroethane	21.2	0	/	139	0	0.004
1,3-dichlorobenzene	37.7	0	/	139	N/A	0.13 U
1,4-dichlorobenzene	0.546	0	/	139	N/A	0.13 U
1,1-dichloroethane	NC	0	/	139	N/A	0.13 U
1,1-dichloroethylene	8.28	0	/	139	N/A	0.13 U
cis-1,2-dichloroethylene	NC	0	/	139	N/A	0.13 U
1,2-dichloropropane	32.7	0	/	139	N/A	0.13 U
cis-1,3-dichloropropene	NC	0	/	139	N/A	0.13 U
ethylbenzene	5.16	0	/	139	0	0.024
2-hexanone	NC	0	/	139	N/A	0.38 U
isopropylbenzene	NC	0	/	139	0	0.035
methyl acetate	NC	0	/	139	0.04	2.6
methyl cyclohexane	NC	0	/	139	0	0.079

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4-methyl-2-pentanone	NC	0	/	139	0.01	0.013
methyl tert-butyl ether	NC	0	/	139	N/A	0.063 U
methylene chloride	4.05	0	/	139	N/A	0.25 U
styrene	4.69	0	/	139	N/A	0.13 U
tetrachloroethylene	9.92	0	/	139	N/A	0.13 U
toluene	200	0	/	139	0.01	0.37
1,2,3-trichlorobenzene	20	0	/	139	N/A	0.13 U
1,2,4-trichlorobenzene	20	0	/	139	N/A	0.13 U
1,1,2,2-tetrachloroethane	0.127	0	/	139	N/A	0.13 U
1,1,1-trichloroethane	29.8	0	/	139	N/A	0.13 U
1,1,2-trichloroethane	28.6	0	/	139	N/A	0.13 U
trichloroethylene	12.4	0	/	139	N/A	0.13 U
1,1,2-trichloro-1,2,2-trifluoroethane	NC	0	/	139	N/A	0.25 U
trichlorofluoromethane	NC	0	/	139	N/A	0.25 U
trans-1,2-dichloroethylene	0.784	0	/	139	N/A	0.13 U
trans-1,3-dichloropropene	NC	0	/	139	N/A	0.13 U
vinyl chloride	0.646	0	/	139	N/A	0.13 U
m/p-xylene	NC	0	/	139	0.01	0.46
o-xylene	NC	0	/	139	0	0.13
acenaphthene	20	0	/	159	0.03	0.63
acenaphthylene	682	0	/	159	0.03	0.21
anthracene	1480	0	/	159	0.1	1.9
benzo[a]anthracene	5.21	1	/	159	0.3	20
benzo[a]pyrene	1.52	3	/	159	0.27	16
benzo[b]fluoranthene	59.8	0	/	159	0.36	23
benzo[g,h,i]perylene	119	0	/	159	0.27	9.3
benzo[k]fluoranthene	148	0	/	159	0.14	10
chrysene	4.73	1	/	159	0.34	20
dibenz[a,h]anthracene	18.4	0	/	159	0.07	2.4
fluoranthene	122	0	/	159	0.48	35
fluorene	122	0	/	159	0.03	0.38
indeno[1,2,3-c,d]pyrene	109	0	/	159	0.17	8.3
naphthalene	0.0994	83	/	159	0.17	1.1
phenanthrene	45.7	0	/	159	0.35	6.2
pyrene	78.5	0	/	159	0.55	35
1-methylnaphthalene	NC	0	/	139	0.15	0.99
2-methylnaphthalene	3.24	0	/	155	0.27	1.6
Low Molecular Weight PAH's with non-detects	NC	0	/	159	0.95	8.85

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High Molecular Weight PAH's with non-detects	NC	0	/	159	2.02	128
Total Extractable Petroleum Hydrocarbons	NC	0	/	136	206	1800
aldrin	0.00332	0	/	141	N/A	0.049 U
alpha-bhc	0.0994	0	/	141	0	0.014
alpha-chlordane	0.224	0	/	141	0	0.014
beta-bhc	0.00398	1	/	141	0	0.01
beta-chlordane	0.224	0	/	141	0	0.012
4,4-DDD	0.758	2	/	141	0.08	7.2
delta-bhc	NC	0	/	141	0	0.019
dieldrin	0.00238	20	/	141	0	0.078
endosulfan I	NC	0	/	141	0	0.0073
endosulfan II	NC	0	/	141	0	0.023
endosulfan sulfate	0.0358	0	/	141	0	0.019
endrin	0.0101	1	/	141	0	0.018
endrin aldehyde	0.0105	14	/	141	0.01	0.11
endrin ketone	NC	0	/	141	0.01	0.14
gamma-BHC	0.005	1	/	141	0	0.0086
heptachlor	0.00598	14	/	141	0	0.021
heptachlor epoxide	0.152	0	/	141	0	0.025
methoxychlor	0.0199	2	/	141	0.02	0.27
toxaphene	0.119	0	/	141	N/A	4 U
aluminum	50	155	/	155	11079	31100
antimony	0.27	130	/	155	1.4	14.3
arsenic	9.9	49	/	160	9.77	64
barium	283	50	/	155	313	2210
beryllium	10	2	/	155	1.08	11.1
cadmium	0.36	92	/	155	0.84	6.58
calcium	NC	0	/	155	88830	376000
chromium	0.4	154	/	155	27.58	110
chromium VI	130	0	/	11	N/A	25.7 U
cobalt	0.14	146	/	155	8.63	34.9
copper	5.4	146	/	155	77.68	408
iron	NC	0	/	155	18496	73500
lead	0.0537	158	/	158	282	2900
magnesium	NC	0	/	155	3618	14900
manganese	220	79	/	155	225	599
mercury	0.1	153	/	176	2.28	31.7
methyl mercury	NC	0	/	20	0	0.0053
molybdenum	2	52	/	139	2.61	19.3

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nickel	13.6	113	/	155	32.96	198
potassium	NC	0	/	155	1838	4980
selenium	0.0276	64	/	155	1.88	9.03
silver	1.04	46	/	155	0.93	5.94
sodium	NC	0	/	155	390	7130
vanadium	2	152	/	155	40.11	165
zinc	6.62	143	/	155	231	1770
thallium	1	3	/	155	0.23	3.01
cyanide (total)	1.33	47	/	123	9.29	120
extractable cyanide	NC	0	/	16	N/A	1.1 U

Models were run for both indicator species using the mean, and 95% upper confidence interval (UCL) values for both sediment (Spotted sandpiper) and soil (American robin) and resulting from samples collected at the site. In addition, effects of "outlier" values on the exposure point concentration were evaluated by eliminating those points and rerunning the model.

The exposure points concentrations for different contaminants evaluated in soil and sediment are included in the results tables.

Sensitivity of Toxicity Reference Values (TRVs)

Given their direct influence on the Hazard Quotient calculation, representing the denominator in Equation 1, the choice of TRVs used in modeling has a major impact on the risk estimate calculated. In many cases TRVs are unavailable for the species being modeled at the site. For example, in the case of modeling impacts to the Spotted sandpiper, there may well be no effects data in the literature for a given contaminant. In such cases, related studies using data from the most taxonomically related species are often used. USEPA will often apply an adjustment factor by dividing the TRV by 10 to account for the uncertainty associated with assuming similar effects to species that are not directly related taxonomically. By attempting to manage uncertainty in this way, it drives risk estimates much higher, and there is no scientific basis for the factor of 10.

USEPA (SSL document) performed comprehensive literature reviews of TRVs for use in developing screening levels for conservative assessment of ecological risks in soil. This approach enables investigators to reduce the uncertainty of risk estimates by focusing on peer-reviewed papers that emphasize effects with known impacts at the population level, such as mortality, reproductive effects, and effects on growth.

However, while the use of the more conservative TRVs are suitable for screening level assessments, review of these documents indicates tremendous variability (often orders of magnitude) in ecotoxicological effects both in studies of the same receptor as well as between receptor species.

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Different toxicity reference values taken from the literature were used for comparison of risk estimates to both the Spotted sandpiper and American robin using sediment and soil data from the example New Jersey site. Tier 1 values were taken from the New Jersey Ecological Evaluation Technical Guidance document (2024) and represent conservative No Observable Adverse Effects (NOAEL) values from the literature. In some cases they reflect species data such as mink that are more sensitive than other species regarding contaminant effects. Tier 2 data are also NOAEL data but reflect values used for development of conservative soil screening values (SSLs) used for screening purposes. So while in many cases these are higher numbers than the Tier 1 values, they are still very conservative and tend to overestimate actual risks. Tier 3 values were selected from the literature, usually the same literature used for development of EPA SSL documents, but are focused on survival, growth and reproduction endpoints to receptors most closely related taxonomically to the Spotted sandpiper and American robin. Hence, generally they are less conservative and more realistic indicators of risk than the Tier 1 or 2 values. The sensitivity analysis was conducted using the 95% UCL of the mean sediment and soil values to which the Spotted sandpiper and American robin, respectively would be exposed at the site.

Sensitivity Analysis of Bioaccumulation Estimates

The terms bioaccumulation factor (BAF) and bioconcentration factor (BCF) are often used interchangeably in the literature, but the latter refers primarily to uptake of contaminants by aquatic organisms directly from the water column. Bioaccumulation Factors in food chain modeling are often single values used to represent the proportion of a contaminant expected to be taken up by a given organism, generally a prey item, that is ingested by the receptor being modeled for risk. In reality, bioaccumulation often follows a curve that plateaus at some point where the organism either dies, or becomes sick and avoids the food item it is ingesting. Differences in BAFs used in modeling can contribute a high degree to the uncertainty of the modeling results.

Bioaccumulation factors were investigated using a single contaminant, lead, as an example of the terrestrial pathway. Site-specific empirical data on earthworm tissue concentrations were used for the food chain modeling. These were derived from a 28-day laboratory investigation of bioaccumulation using “red worms” *Eisenia foetida*, a commonly used earthworm species in laboratory testing and bioaccumulation studies.

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Table 4. *Comparison of Soil Concentrations of Various Contaminants of Concern to Earthworm Concentrations Measured in Tissue at the end of a 28-day Laboratory Test*

ANALYTE	Soil Concentration Maximum (mg/kg DW) n = 26	Soil Concentration Mean (mg/kg DW) n = 26	Earthworm Tissue Concentration Maximum (mg/kg DW) n = 0-14	Earthworm Tissue Concentration Mean (mg/kg DW) n = 0-14	BAF based on Maximum Concentration	BAF based on Mean Concentration
SEMI-VOLATILE ORGANIC COMPOUNDS						
benzo[a]anthracene	20	0.31	0.04	0.033	0.002	0.11
benzo[a]pyrene	16	0.29	0.016	0.0063	0.001	0.022
chrysene	20	0.37	0.15	0.071	0.0073	0.19
naphthalene	1.1	0.18	0.0079	0.0073	0.0071	0.04
PESTICIDES						
4,4-DDD	4.9	0.33	0.96	0.092	0.19	0.27
4,4-DDE	0.94	0.047	0.1	0.022	0.1	0.47
4,4-DDT	1.4	0.063	0.045	0.007	0.032	0.11
dieldrin	0.2	0.013	0.004	0.0017	0.02	0.13
endrin aldehyde	0.2	0.018	0.015	0.0015	0.074	0.083
heptachlor	0.07	0.0081	0.0001	0.000095	0.0014	0.012
METALS AND ORGANICS						
arsenic	64	10	3.9	1	0.061	0.097
barium	2210	310	5.1	2.6	0.0023	0.0085
beryllium	11.1	1.1	0.021	0.016	0.0019	0.015
cadmium	14	0.9	2.3	0.63	0.17	0.67
copper	408	78	8.4	4.1	0.02	0.052
lead	1700	275	35.4	4.9	0.021	0.018
mercury	32	2.2	0.34	0.16	0.011	0.07
methyl mercury	0.0053	0.002	N/A	N/A	NC	NC
nickel	198	33	1.4	0.56	0.0072	0.017
selenium	9	3.4	1.1	0.8	0.12	0.24
silver	5.9	1.32	0.086	0.027	0.014	0.02
vanadium	165	41	0.71	0.35	0.0043	0.0086
zinc	1770	229	24.5	17.5	0.014	0.076
Notes:						
mg/kg = milligrams/kilogram						
DW = Dry weight						
BAF = Bioaccumulation Factor derived from data (tissue concentration dry weight/soil concentration dry weight)						

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The resultant tissue concentrations in the worms were used to calculate a 95% upper confidence interval (UCL) used for modeling uptake of lead. The BAF was compared to literature-based data on bioaccumulation factors, which vary widely, to investigate the degree of uncertainty involved in typical risk assessments. Different chemicals will have different uptake kinetics, and hence their ability to bioaccumulate will vary. But even using a single contaminant such as lead can illustrate how variable results are on risk assessment findings.

Toxicity reference values were all taken from the literature and include both the No Observable Adverse Effects Level (NOAEL), the concentration below which adverse effects are not seen, and Lowest Observable Adverse Effects Level (LOAEL), the lowest concentration in the literature at which effects are seen. Variability and uncertainty are introduced when the decision is made regarding what receptor to use, and how conservative a value should be chosen to estimate risks. For example, Tier 1 values used by the State of New Jersey are based on agency consensus reflecting highly conservative effects levels so that risk decisions do not result in remaining ecological risks. Tier 2 values represent U.S. Environmental Protection Agency values used to develop Soil Screening Levels (SSLs) that are conservative values used to protect sites, but in many cases are less conservative than Tier 1 values. Tier 3 values represent more realistic values reflecting potential ecotoxicological effects to receptors more closely related taxonomically to the two indicator species investigated.

Data used for modeling soil and sediment concentrations of selected chemicals were taken from an actual manufacturing site in New Jersey, USA. The contaminants chosen were based on those exceeding ecologically based screening benchmarks (USEPA soil screening levels, and New Jersey Department of Environmental Protection Sediment Criteria).

Literature-based values are often used as a basis for calculating the amount of a contaminant present in the tissue. This leads to significant uncertainty and usually an overestimate of the amount of risks present.

Table 5 provides a summary of literature-based values for bioaccumulation of lead from soil-to-soil invertebrates, in comparison to empirically derived data from a site in New Jersey. There is a considerable literature base on bioavailability and uptake of lead into soil invertebrates, and the intent is not to review it here. The focus is on how the variability in bioaccumulation factors reported can influence an ecological risk assessment.

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Table 5. *Comparison of Bioaccumulation Factors in the Literature to Empirically Derived Data from a Site in New Jersey*

Bioaccumulation Factor	Source	Comments
Pb log (worm) = 0.74 log (soil) + 0.05	Neuhauser et al. 1995	Indicates relationship is not linear.
Pb Concentration in worm/Pb Concentration in Soil = $14.45*10^{0.9 \log (\text{Soil Pb})}/10$	ERD, USACE Table 1 (1999)	Indicates relationship is not linear.
0.01 to 22, median =0.23	Oorts et al (2021)	Reviewed 248 papers of bioaccumulation of lead in earthworms.
3.34 (average of 6 papers), range was 0 to 228. Median value = 0.23	Sample et al (1999)	Best predictive equation (r^2 of 0.78 was based on soil Pb, soil calcium and pH. The 228 number strongly suggests an outlier, since most values in the literature they cited were less than 1.5.
0.05 to 5	Kavehei (2017)	Figure 5.3 of his thesis reviewed results of 12 papers and presented 4 results from his study based on different forms of lead (PbO, $PbCl_2$, $PbCO_3$, and PbS).
0.018	Site-specific number, NJ	Sample size was 7 composite worm samples each of 10 worms, and compared to soil from which they were exposed. See Table 6.

Three different BAF values were used for comparison of model results.

- A low-range value from the literature based on the empirically derived model presented in Sample et al (1999) which predicts concentrations of lead in earthworm tissue based on the soil lead concentration, soil calcium concentration and pH.
- A mid-range literature value (0.23) based on the median value from the Oorts et al (2021) paper;
- A high-range value based on the maximum value (5) reported in Kavehei (2017).

Sensitivity Analysis of Home Range Size and Other Inputs

The home range size used for food chain modeling of an indicator species can have an important impact on the model results. For some species, such as the Spotted sandpiper, home ranges have not been well studied outside of breeding colonies, leading to uncertainty in the results. This species may nest solitarily, or in colonies (Cornell 2024). While territory is defined as any defended area (Hinde 1956), home range refers to the 'that area traversed by

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the individual in its normal activities of food gathering, mating, and caring for the young" (Burt 1943). In birds that exhibit territorial behavior, territory size often reflects population size and resource availability (Stenger 1958, Brown 1969).

Because territory is considered to be a subset of home range size in most cases, use of data on territory size as an estimate of home range size would result in an overestimate of ecological risks from contaminant exposure. This is because if the chemical were present in a discrete area, the smaller the home range of the indicator species, the greater likelihood that it would receive a larger dose of the contaminant if it were living on or near the site.

In the case of the Spotted sandpiper, the standard EPA reference, the Wildlife Exposure Factors Handbook (1993) lists only an estimated territory size taken from a breeding colony and provides no data on home range sizes. In this paper we present literature sources representing several different home range sizes, some from different geographic locations (**Table 6**).

Table 6. *Territory and Home Range Sizes of Spotted Sandpipers from the Literature*

Territory or Home Range Size	Reference	Comments
0.25 acres	USEPA Exposure Factors Handbook (1993)	Territory size apparently derived from a breeding colony in Minnesota (reported by Maxson and Oring 1980).
12 acres (mean value)	Miller and Miller 1948	Cited in Maxson and Oring 1980
2.5 acres 4.5 acres 8 acres	Hays 1972	Individually marked female birds from a nesting colony in New York
10 individuals per acre when breeding	Oring et al 1983	Nesting colony on a small island in Minnesota (average breeding density over 9 years)

The different home range sizes were derived from papers reporting the population densities of nesting birds in colonies in Minnesota, California and New York. These are conservative from the perspective of risk estimation because they do not represent the entire foraging radius of individuals but rather the density of nests presents in the colony.

To estimate potential ecological risks different home range sizes ranging from the 0.25-acre territory size to a 12-acre home range were used as inputs to food chain models for the Spotted sandpiper. The risk assessment example was a 1.08-acre pond present at a New Jersey manufacturing facility. To estimate the extent of the area used by the birds, the area of the pond was divided by the home range to obtain an area use factor (AUF). A separate seasonal use factor

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was used conservatively to indicate the birds were present only during breeding season.

Results

Sensitivity of Different Exposure Point Concentrations

Tables 7 and 8 present results of the food chain modeling using different exposure point concentrations as measures of risk to the Spotted sandpiper and American robin, respectively. Backup spreadsheet data for all calculations have been provided in **Appendix 1**.

As can be seen in the results, using the 95% UCL results in a higher risk estimate than that using the mean.

Table 7. *Comparison of Hazard Quotients Calculated with Mean and 95% UCL for American robins*

Hazard Quotients Calculated with Mean and 95% UCL for American Robin					
COPEC	**Mean Concentration in Site Soil (0-6") (mg/kg-DW) n = 26**	**95% UCL in Site Soil (0-6") (mg/kg-DW) n = 26**	**LOAEL (mg/kg BW-WW day) (k)**	**Mean Hazard Quotient (HQ) (unitless)**	**95% Hazard Quotient (HQ) (unitless)**
HMW PAHs	0.329	0.592	0.48	0.05	0.05
Benzo(a)anthracene	0.313	1.37	0.48	0.02	0.03
Chrysene	0.369	1.13	0.48	0.03	0.04
4,4'-DDD	0.334	0.46	2.27	0.01	0.07
4,4'-DDT	0.0629	0.0862	2.27	0	0
Endrin aldehyde	0.0179	0.0121	0.3	0	0
Arsenic	10.02	13.09	2.24	0.11	0.15
Barium	309.7	351.8	41.7	0.12	0.13
Copper	77.7	99.53	4.7	0.33	0.42
Lead	266.1	303.3	1.9	1.91	2.56
Mercury	2.223	3.496	0.9	0.07	0.1
Methyl mercury	0.00198	0.00273	0.026	0.04	0.05
Nickel	32.63	44.01	8.16	0.09	0.13

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Table 8. *Comparison of Hazard Quotients Calculated with Mean and 95% UCL for American robins*

Hazard Quotients Calculated with Mean and 95% UCL for American Robin					
COPEC	**Mean Concentration in Site Soil (0-6") (mg/kg-DW) n = 26**	**95% UCL in Site Soil (0-6") (mg/kg-DW) n = 26**	**LOAEL (mg/kg BW-WW day) (k)**	**Mean Hazard Quotient (HQ) (unitless)**	**95% Hazard Quotient (HQ) (unitless)**
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Chrysene	0.369	1.13	0.48	0.03	0.04
4,4'-DDD	0.334	0.46	2.27	0.01	0.07
4,4'-DDT	0.0629	0.0862	2.27	0	0
Endrin aldehyde	0.0179	0.0121	0.3	0	0
Arsenic	10.02	13.09	2.24	0.11	0.15
Barium	309.7	351.8	41.7	0.12	0.13
Copper	77.7	99.53	4.7	0.33	0.42
Lead	266.1	303.3	1.9	1.91	2.56
Mercury	2.223	3.496	0.9	0.07	0.1
Methyl mercury	0.00198	0.00273	0.026	0.04	0.05
Nickel	32.63	44.01	8.16	0.09	0.13
Vanadium	40.8	48.4	0.688	0.38	0.46

Table 9. *Comparison of Hazard Quotients Calculated with Mean and 95% UCL for Spotted sandpipers*

COPEC		**Mean Concentration in Site Sediment (0-6") (mg/kg-DW) n=0-36**	**95% UCL in Site Sediment (0-6") (mg/kg-DW) n=0-36**	**LOAEL (mg/kg WW-BW day) (l)**	**Hazard Quotient (HQ) (unitless)**	**Hazard Quotient (HQ) (unitless)**
HMW PAHs		1.081	1.555	0.48	0.03	0.04
2-Methylnaphthalene		0.256	0.29	6.7	0	0
Acenaphthene		0.043	0.043	6.7	0	0
Acenaphthylene		0.0848	0.0848	6.7	0	0
Anthracene		0.143	0.22	6.7	0	0

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Benzo(a)anthracene	0.103	0.122	0.48	0	0
Benzo(a)pyrene	0.109	0.139	0.48	0	0
Benzo(g,h,i)perylene	0.231	0.303	0.48	0.01	0.01
Chrysene	0.307	0.491	0.48	0.01	0.01
Dibenz(a,h)anthracene	0.111	0.131	0.48	0	0
Fluoranthene	0.141	0.218	0.48	0	0.01
Fluorene	0.04	0.04	6.7	0	0
Indeno(1,2,3-c,d)pyrene	0.127	0.128	0.48	0	0
Naphthalene	0.245	0.286	6.7	0	0
Pyrene	0.31	0.454	0.48	0.01	0.01
4,4'-DDD	5.431	9.305	0.027	28.5	45.56
4,4'-DDE	0.764	1.103	0.027	1.5	1.67
Arsenic	47.89	54.44	3.55	0.22	0.25
Barium	505.52	554.5	41.7	0.12	0.13
Cadmium	5.393	6.444	1.47	0.14	0.16
Chromium	83.15	93.21	2.66	0.44	0.49
Cobalt	10.5524	11.65	7.8	0.07	0.08
Copper	216.5	247.1	4.7	3.73	4.25
Lead	1718	1955	1.9	11.51	13.1
Mercury	3.931	4.374	0.026	9.35	10.39
Methyl mercury	0.0141	0.0251	0.026	0.07	0.1
Nickel	40.48	43.99	6.71	0.19	0.21
Selenium	5.084	5.084	0.29	0.79	0.79
Silver	3.935	4.33	2.02	0.03	0.04
Vanadium	141.9	165.5	0.688	2.01	2.35
Zinc	432.5	492.3	66.1	0.66	0.75

Sensitivity Analysis of Different TRV Values

Table 10 and 11 show the influence of using different TRV values on the estimate of risks to Spotted sandpipers and American robins, respectively. Hazard quotients greater than 1 denote ecological risks.

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Table 10. *Sensitivity Analysis of Different TRVs on Risk Estimate for the Spotted Sandpiper*

COPEC	Tier 1 TRV (mg/kg BW day)	HQ	Tier 2 TRV (mg/kg BW day)	HQ	Tier 3 TRV (mg/kg BW day)	HQ
HMW PAHs	0.48	0.035	Not Available	NC	20	0.00084
4,4'-DDD	0.027	46	2.27	0.54	0.028	44
4,4'-DDE	0.027	1.7	2.27	0.020	0.028	1.6
Copper	4.7	4.3	12.1	1.7	61.7	0.32
Lead	1.9	13	3.26	7.6	11.3	2.2
Mercury	0.026	10	Not Available	NC	0.90	0.30
Methyl mercury	0.026	0.10	Not Available	NC	0.064	0.039
Zinc	Not Available	NC	66.1	0.75	131	0.38

Table 11. *Sensitivity Analysis of Different TRVs on Risk Estimate for the American robin.Sensitivity Analysis of Different Bioaccumulation Factors*

COPEC	Tier 1 TRV (mg/kg BW day)	HQ	Tier 2 TRV (mg/kg BW day)	HQ	Tier 3 TRV (mg/kg BW day)	HQ
HMW PAHs	0.48	0.048	Not Available	NC	20	0.0012
4,4'-DDD[1]	2.27	0.065	2.27	0.065	0.028	5.3
4,4'-DDT[1]	2.27	0.0014	2.27	0.0014	0.028	0.11
Copper	4.7	0.42	12.1	0.16	61.7	0.032
Lead	1.9	2.6	3.26	1.5	11.3	0.43
Mercury[2]	0.90	0.10	Not Available	NC	0.9	0.10
Methyl mercury	0.026	0.052	Not Available	NC	0.064	0.021

Tables 12 shows the influence of using different bioaccumulation factors (BAFs) on calculated tissue concentrations in earthworm, using lead as an example contaminant of concern, and earthworms as a typical prey item for American robins. These results were compared to the empirical site-specific data obtained from a 28-day bioaccumulation study of soil present on site, measured along a contamination gradient.

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Table 12. *Earthworm Tissue Concentrations Measured Empirically versus Estimated by a Range of Bioaccumulation Factors*

95% UCL of Lead in Site Soil (0-6") (mg/kg-DW) n = 26	Bioaccumulation Factor (BAF) (Soil to Earthworms)	Estimated Concentration in Earthworms (mg/kg-DW)
303.30	Empirically Measured 95% UCL	9.52
303.30	Empirically Measured Mean	4.9
303.30	0.23 (median of Oorts et al 2021)	70
303.30	5.00 (selected from papers revised by Kavehei 2017)	1516
303.30	Sample et al 1999 Equation	3.71

Sample et al (1999) equation: ln(earthworm)= B0 + B1(ln[soil]) +B2(ln[soil Ca])+ B3(pH)
where B0=intercept, B1=soil coefficient, B2=soil Ca coefficient, and B3= pH coefficient.
B0 = 2.45, B1= 1.18, B2= 0.06, B3= -.93

In the case of American robins, tissue data collected from the site were used to estimate the concentration of contaminants being ingested by the birds. Focusing on lead, approximately 3% of the lead in soil was present in depurated tissue of the earthworms. Site-specific values can provide a much more accurate estimate of the bioavailability of the soil contaminants present, in this case lead. Actual prey item tissue concentrations may be used in place of a bioaccumulation factor to estimate risks. It should be noted that collection of empirical tissue data may be more expensive and involves careful study design to ensure that the entire soil contamination gradient is assessed. **Table 13** shows the effect of using different BAFs on risk estimates for the American robin.

Table 13. *Results of Food Chain Modeling for American Robin based on Different Bioaccumulation Factors*

Bioaccumulation Factor	Earthworm Concentration (95% UCL), Measured or Modeled	Hazard Quotient to American robins
Empirically Measured 95% UCL	9.52	2.56
Empirically Measured Mean	4.9	2.12
Median of Oorts et al (2021) =0.23	70	8.26
Selected from Kavehei 2017 as higher value = 5	1516	145
Sample et al 1999 Equation	3.71	2.01

Results of the analysis indicate that the Sample et al (1999) model came close to predicting the actual earthworm tissue lead concentration if the mean lead concentration (not 95% UCL) value is used. The effect of using the

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modeled concentration resulted in similar risk estimates for the American robin as the hazard quotients are both near 2.0. Using other literature-based numbers, however, greatly overestimated risks.

Sensitivity Analysis of Home Range Size on Risk Estimates for Spotted Sandpiper

Table 14 provides the results of the food chain modeling with different home range sizes from the literature.

Table 14. *Results of the Sensitivity Analysis of Home Range Size on Hazard Quotients Calculated for Spotted Sandpipers*

COPEC	95% UCL	Area Use Factor[(a)]				Hazard Quotient			
	Concentration in Sediment	0.25 acre	2.5 acre	4.5 acre	12 acre	0.25 acre	2.5 acre	4.5 acre	12 acre
4,4'-DDD	9.305	1	0.432	0.24	0.09	45.56	19.68	10.93	4.1
4,4'-DDE	1.103	1	0.432	0.24	0.09	1.67	0.72	0.4	0.15
Copper	247.1	1	0.432	0.24	0.09	4.25	1.84	1.02	0.38
Lead	1955	1	0.432	0.24	0.09	13.1	5.66	3.14	1.18
Mercury	4.374	1	0.432	0.24	0.09	10.39	4.49	2.49	0.94
Methyl Mercury	0.0251	1	0.432	0.24	0.09	0.1	0.04	0.02	0.01
Zinc	5.084	1	0.432	0.24	0.09	2.35	0.32	0.18	0.07

[(a)]Calculated by taking 1.08-acre Pond Area and dividing by Home Range

Discussion

Exposure Point Concentrations

In the food chain modeling conducted of both the Spotted sandpiper and American robin, use of the 95% UCL as the exposure point concentration gave higher estimates than that of the mean. This is not surprising, given that the 95% UCL is affected by higher values in the distribution, and contaminant data are often lognormally distributed. In come cases these higher values represent "hot spots" that may be addressed by remediation. But there is always considerable uncertainty regarding the exposure point concentration because it is affected by sample size, variability in the soil matrix, and in some cases the spotty nature of contaminant distribution in fill material and other soil substrates affected by atmospheric deposition and other sources. Moreover, analytical data from soil or sediment represent a very small sample volume, as low as 0.5 g of material (USEPA Method SW-846). An individual analytical result is then used to represent the contamination found at a location that may

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be 10 m in radius or more. Resampling or duplicate samples can provide an indication of the variability associated with sampling results but is expensive and typically quality assurance and control procedures are based on 1 out of 10 samples being duplicated. The variability can be substantial and significantly affect risk estimates.

Consider the following table of results obtained when conducting the earthworm bioaccumulation study cited above.

Table 15. *Comparison of Maximum Analytical Data Originally Selected for Testing Versus Concentrations Detected in Actual Soil Samples Used for the Earthworm Bioaccumulation Study*

Analyte	**Location**	**Prior Maximum Concentration (Jan 2018)**	**Resampled Concentration (Oct-Dec 2018)**
Unit		**mg/kg**	**mg/kg**
4,4'-DDD	CSB-645	2.3	0.12J
Toxaphene	CSB-645	98	ND
Arsenic	CSB-647	143	32.4
Barium	CSB-647	5210	121
Chromium	CSB-670	611	76.6
Copper	CSB-666	700	150
Lead	ESB-489	2900	501
Mercury	ESB-520	31	22.8
Vanadium	CSB-546	2090	36.3
Zinc	ESB-447	1770	1500
Cyanide	ESB-514	120	6.6

This paper focused on lead, and review of the maximum concentration intended to be measured indicates when the same location was resampled a few months later, a much lower concentration was detected. This cannot be explained by attenuation or any other natural process, but rather reflects the random variability associated with sampling a second time at the same depth. This phenomenon was not confined to lead concentrations; while some parameters such as zinc showed similar results between events, many others such as 4,4'-DDD, arsenic, barium, chromium, and copper were detected at significantly lower values during the second event. This variability is significant, and the earthworm study were not conducted, and the original soil data were used along with literature-based bioaccumulation factors, a significantly higher risk estimate would have resulted.

Analytical methods can also affect results and may have been a factor in the above results. Typically, analytical methods such as EPA SW-846 of metals require reproducibility of matrix spikes to fall within 20% of the known matrix spike in order for results to meet QA/QC criteria (USEPA 2024). With respect to organics, the criterion is often 30%. With such variability in analytical

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results, risk estimates could vary significantly as a result of analytical testing alone. If analytical variability and sampling variability are both accounted for, the resulting variability could result in order of magnitude differences in risks calculated to indicator species. Yet risk assessments seldom calculate this range quantitatively.

Toxicity Reference Values

One of the greatest sources of uncertainty in food chain modeling may well be from the choice of TRV in calculating risks. Values are often suggested or required by regulatory agencies to allow conservative decisions to be made regarding whether a legacy site may require remediation. However, in some cases the assessments are made based on singe research papers that are decades old and which have not been repeated experimentally to achieve the same or similar results. Many times, these studies are entirely laboratory based and the measure of effects is based on experimental feeding trials or dosage studies that use highly available forms of contaminants that may not represent the same forms that would be found in nature within sediments or soil. Highly bioavailable forms are used so that researchers can see at what concentrations effects are evident, as opposed to studies of soil or sediment from the field that may show little or no effect.

Yet, conducting food chain modeling based on expensive, carefully measured empirical data collected from the field on sediment, soil and tissue and then using it to calculate a dose that is then compared to a single laboratory-based study is problematic. In-situ field tests in which an organism is exposed to soil or sediment under natural conditions may be a solution, but the organism should be one that would normally occur at the site and acclimated to site conditions. Such tests often fail for not meeting those criteria. For example, exposing earthworms such as *Eisenia foetida* (characteristic to European dung piles) may not be appropriate in evaluating sandy soils contaminated with lead in the U.S. The animals could react adversely to low pH, grain size or lack of organic matter necessary for food during a 28-day test, for example.

In this current study the choice of TRV clearly affected risk estimates. Although results would still indicate the need for remedial action in some cases if a lower TRV were chosen, the results would be different if a different data set were applied (e.g. at lower soil concentrations of 4,4’ DDE to American robins for example.

Bioaccumulation Factors

Risk estimates to American robins were highly influenced by the choice of bioaccumulation factors used to determine the concentration of lead in earthworm tissue. The hazard quotients used to estimate risk varied by over an order of magnitude. These estimates would be even higher if a receptor that fed entirely on earthworms were modeled, such as a shrew, or American

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woodcock. While it varies seasonally, the American robin's diet is approximately half vegetative material (Wheelwright 1988). If a receptor were modeled that ingested almost exclusively earthworms, these hazard quotients would be nearly twice as high. Moreover, the hazard quotients calculated were based on a site with moderate lead contamination (e.g. approximately 300 mg/kg dry weight on average) in surficial soils, which is much lower than some mining or manufacturing sites where concentrations may be in the thousands of parts per million.

Risk estimates calculated using the literature based BAFs would be even more overestimated if the soil data were higher because of the direct relationship between soil concentration and earthworm concentrations in the model. Lead bioaccumulation is not necessarily linear and hence at higher soil concentrations the amount in tissue would be overestimated by a single BAF.

Several papers have specifically focused on uptake of lead by various species of earthworms in laboratory bioaccumulation tests (Bradham et al 2006, Sandifer et al, 1996, Kiewet et al, 1991). These, and other investigations cited in **Table 5** of this paper indicate a high degree of variability based on soil properties with BAFs ranging from 0.01 to 22.05.

Why would bioaccumulation factors in the literature from soil to earthworms vary so much? The potential factors include:

1. Experimental design.
2. Form of lead present.
3. Soil characteristics affecting bioavailability, including pH, calcium concentrations, organic matter, and grain size.
4. Amount that the worms are depurated.

Bioaccumulation factors cited in the literature that are far greater than 1 are likely an overestimate of the actual amount present in earthworm tissue for a chemical such as lead, which although it can be toxic is not considered a strong bioaccumulator in terrestrial food chains (ATSDR 2007, Eisler 1988). According to Eisler (1988), lead poisoning in higher organisms has been associated with lead shot and organolead compounds, but not with food chain exposure to inorganic lead in soil (other than direct ingestion of lead shot, sinkers or paint).

Calculated BAFs such as the 228-maximum cited in Sample et al (1998) appear to be outliers may reflect situations where the earthworms were not fully depurated prior to analysis. Such BAFs should not use in risk assessment modeling. Neither should BAFs that are based on laboratory studies of earthworm uptake by forms of lead that are highly bioavailable, but not often found in nature.

Regardless of whether bioaccumulation is measured in field or laboratory studies, depuration is critical to standardization of results. Why depurate earthworms as a basis for risk estimates? Because the soil fraction is already accounted for in the models, and the soil fraction itself is overestimating risks because it is not 100% bioavailable. In fact, because there is no adjustment in

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the model for the bioavailability of lead in soil, the soil fraction often drives the estimate of risk. The estimate is biased high because typically only a portion of the lead in soil may be assimilated in a form that is mobilized upward through the food chain. While it is true that birds would consume the soil inside the earthworm and not depurate it first, it is the portion in tissue that is most bioavailable and hence likely to be passed upward through the food chain. Ingestion of soil lead can be mitigated by diets high in protein and calcium which tend to inhibit absorption in the digestive system (reviewed in Franson and Paine 2011, p.568).

As Sample et al (1998), site-specific studies of bioaccumulation are nearly always more accurate than literature-based results. Site-specific bioaccumulation may be measured two different ways. In the case of earthworms, the first approach is to collect worms at the site and depurate them. The drawback to this approach is it is limited to where worms are present. Highly contaminated areas or locations with poor soil characteristics (coarse grain size, little organic matter) may not be supporting sufficient worm populations to sample. In addition, it may be difficult to relate the resultant tissue concentrations back to the soil concentration. For example, suppose it takes a meter- squared plot to obtain sufficient biomass to measure tissue concentrations. Establishing a mean soil concentration over such a large area may require several surficial samples to be collected and analyzed. A single composite sample may not reflect concentrations to which the worms are exposed.

A second approach would be to look at existing soil analytical data to determine a sampling gradient. Those locations can be resampled to confirm the gradient and expose the worms to sufficient soil in the laboratory to obtain estimates of uptake over a 28-day period. The actual worm tissue data are then used to calculate risks. If risks are observed, the empirically derived BAFs can be used to back calculate the concentration in soil that is “safe” as a basis for remedial action recommendations. A drawback of this approach is the original contaminant gradient observed may not be reproducible (see Table 12 above for an example. For that reason, it is a good practice in designing such studies to include “extra” or duplicate locations to help ensure that the worms analyzed reflect the range of concentrations present at the site.

Conclusions and Recommendations

This paper has focused on many sources of uncertainty in ecological risk assessments that have been known about for years but are seldom quantified to evaluate their impact on risk estimates. Specifically, food chain modeling relies on several critical assumptions that may significantly affect risk estimates.

These include exposure point concentrations calculated for soil and sediment, literature-based bioaccumulation factors, toxicity reference values from most sensitive species, and home range sizes assessed from limited data. In example calculations from a site in New Jersey the variability in such inputs resulted in differences in risk estimates to the Spotted sandpiper and American robin that sometimes varied over an order of magnitude.

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Specific recommendations to improve the reliability of risk estimates used for remedial action decisions included the following.

1. Re-evaluate or replicate older (e.g. decades old) toxicity studies that are still used as a basis for many risk assessments to verify their validity.
2. Conduct more toxicological studies on wildlife species as opposed to common laboratory animals (e.g. rats, mice) that then require extrapolation to wild animals.
3. Avoid applying TRVs that were developed as a basis for screening benchmarks to risk assessments used as a basis for remedial action decisions.
4. Ensure that indicator species modeled inhabit or can inhabit the site, and that AUFs calculated reflect the actual extent that the species could use the site.
5. Regulatory agencies would benefit from developing databases of site-specific studies conducted at legacy sites nationwide for sharing information on risks. This “gray literature” (not peer-reviewed or in journals) could provide very useful information on actual site-specific risks that has not been reported in the literature that is dominated by (often older) laboratory-based toxicity studies.

Implementation of regulatory policies designed to obtain more realistic risk estimates could allow refocusing regulatory efforts, including research and funding toward other initiatives such as habitat preservation via conservation easements, and land acquisition as part of the risk management process. This could result in more positive ecological benefits than remediation of sites that may not actually require it based on overestimates of risks.

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